Nitrogen Risk Assessment Model for Scotland: I. Nitrogen leaching

S. M. Dunn, A. J. A. Vinten, A. Lilly, J. Degroote, M. A. Sutton, M. Mcgechan

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S.M. Dunn[1], A.J.A. Vinten[2], A. Lilly[1], J. DeGroote[1], M.A. Sutton[3] and M. McGechan[2]

[1]Macaulay Institute, Craigiebuckler, Aberdeen, AB15 8QH, Scotland
[2]Scottish Agricultural College, Bush Estate, Penicuik, Midlothian, EH26 OPH, Scotland
[3]Centre for Ecology and Hydrology, Bush Estate, Penicuik, Midlothian, EH26 0QB, Scotlan

E-mail for corresponding author: s.dunn@macaulay.ac.uk

Abstract

The Nitrogen Risk Assessment Model for Scotland (NIRAMS) has been developed for prediction of streamwater N concentrations draining from agricultural land in Scotland. The objective of the model is to predict N concentrations for ungauged catchments, to fill gaps in monitoring data and to provide guidance in relation to policy development. The model uses nationally available data sets of land use, soils, topography and meteorology and has been developed within a Geographic Information System (GIS). The model includes modules to calculate N inputs to the land, residual N remaining at the end of the growing season, weekly time-series of leached N and transport of N at the catchment scale. This paper presents the methodology for calculating N balances for different land uses and for predicting the time sequence of N leaching after the end of the growing season. Maps are presented of calculated residual N and N leaching for the whole of Scotland and the spatial variability in N leaching is discussed. The results demonstrate the high variability in N leaching across Scotland. The simulations suggest that, in the areas with greatest residual N, the losses of N are not directly proportional to the amount of residual N, because of their coincidence with lower rainfall. In the companion paper, the hydrological controls on N transport within NIRAMS are described, and results of the full model testing are presented.

Keywords: nitrogen, diffuse pollution, agriculture, leaching, land use, model, national, catchment

Introduction

Implementation of EU legislation such as the Nitrates and Water Framework Directives (EC, 1991 and EC, 2000) requires an extensive understanding of the hydrochemical quality of surface and groundwater resources. Generally, monitoring of water bodies provides data for larger river systems with limited coverage of groundwaters. There is often a lack of data pertaining to water quality in ungauged basins, including sub-catchments and small streams that drain directly to the sea. In response to the perceived need for this information, the Scottish Executive commissioned a project to develop a model to predict concentrations and fluxes of surface and groundwater nitrates draining from agricultural land, to be applicable anywhere in Scotland. This and the companion paper (Dunn *et al*., 2004) together describe the development and preliminary testing of the resulting NIRAMS (Nitrogen Risk Assessment Model for Scotland) model.

Marsden and MacKay (2001) consider diffuse agricultural pollution to be the second most important source of pollution (after sewage effluent) in Scottish rivers and lochs. One of the major diffuse pollutants that is sourced from agriculture is nitrate which, due to its high solubility, is easily leached from the soil to both ground and surface waters.

Gradual increases in streamwater nitrate concentrations have occurred over the last 50 years, and have been associated with the application of organic and inorganic nitrogenous fertilisers to agricultural land (SEPA, 1999). In some Scottish waters, nitrate concentrations have reached the EU drinking water limit of 11.3 mg l^{-1} and some estuarine waters have been identified as eutrophic (SEPA, 2000). These changes have led to recent designation of large areas in the east of Scotland as Nitrate Vulnerable Zones (SEERAD, 2003).

The development of large-scale spatial models that highlight the sources, distribution and transport of nitrates can assist in the creation of future management plans for controlling diffuse pollution. Many models have been developed to predict nitrate leaching and transport at catchment scales e.g. INCA (Whitehead *et al*., 1998), SWAT (Neitsch *et al*., 2002) and AGNPS (Young *et al*., 1989), but these models are complex and have detailed data requirements. As such, they are not useful screening tools for use in a planning context such as that required by the EU Nitrates Directive. At the other extreme, very simple models, such as those based on export coefficients (e.g. Johnes, 1996), do not give adequate information about either spatial or temporal variability in catchment scale N processes. The SPARROW model (Smith *et al*., 1997) developed a regression of N transport based on a range of catchment characteristics. Agriculture was included in a simplistic manner using fertilizer application and livestock waste production as the two main parameters. This model provides a simple methodology for estimating N pollution on a national scale but is limited in its potential for investigating impacts of land use change or temporal variability in stream water quality. In England and Wales, the MAGPIE model (Lord and Anthony, 2000) was developed as a modelling framework to evaluate nitrate losses at the national scale; it provides a good example of integration of nationally available datasets to predict nitrate leaching. The objective of the NIRAMS approach was to provide a similar tool to MAGPIE that could be applied within Scotland. The structure and detail of nationally available agricultural and biophysical data for Scotland is different to England and Wales, although similar types of information exist. Also, differing agro-climatic conditions in Scotland affect typical farming practices making a direct application of MAGPIE to Scotland inappropriate. In addition to this, NIRAMS has been designed to include a more explicit budgeting of N at a local scale and a more comprehensive methodology for hydrological transport and routing to predict streamwater N concentrations.

In developing a model to be applicable at the national scale and to predict water quality in ungauged catchments, many factors have to be considered, including the necessity for input data and parameters to be derived from nationally available data sets, limited calibration requirements and the ability to model catchments with very diverse characteristics. For NIRAMS, several core spatial datasets were identified to provide uniform under-pinning definitions of topography, soil and land cover distributions. These datasets formed the basis of the subsequent model. Wherever possible, parameter values would be linked to the core datasets, leaving only a limited number of parameters to be fitted by model calibration. To provide a flexible system that could be applied readily to user-defined sub-catchment or catchment units, the model was developed fully within a Geographic Information System (GIS). The land is represented in NIRAMS by raster cells of 1 km^2 resolution, with leaching calculations carried out for each cell, on the basis of its physical properties and management.

The major source of complexity in existing N models occurs because of their attempts to model all of the processes involved in the N cycle. This is necessary to predict detailed temporal variability in the amount of N available in the soil for leaching at any time. An alternative approach that provides a major simplification is to calculate annual N balances of the inputs and outputs to the N cycle, including gaseous fluxes, and from these to determine how much excess N (if any) has been added to the soil over a year. Because of the seasonal nature of plant uptake and soil wetness, a simple assumption is that excess N accumulated during a year as a result of agricultural inputs begins to leach from the soil only at the end of the growing season. As soils return to field capacity, drainage begins and a large proportion of the residual N is lost during the winter period (Burns, 1980). The timing and amount of leaching is then controlled largely by hydrological conditions, rather than by specific processes of the N cycle. This simplification provides a methodology appropriate for large scale modelling and was therefore adopted as the basis of the NIRAMS model. The growing season in Scotland is assumed to stop at the end of August, after which time excess residual N can be lost progressively to leaching. Calculations of the leaching are made using a time-step of one week. In practice, leaching does not generally begin until around October, when the soil has wetted up sufficiently.

The different stages involved in transporting N from the land into the freshwater system, can be divided into a series of steps involving:

(1) Inputs of N to the land or water (from the atmosphere, agriculture or urban / industrial inputs);
(2) N cycling in the soil system;
(3) Leaching of residual N remaining in the soil at the end of the growing season;
(4) Transport of leached N to surface and groundwater systems.

Each of these steps is represented by a different module in NIRAMS (Fig. 1). Both the leaching and transport of N are linked strongly to hydrological processes and additional hydrological modules have been developed to describe these. Within this paper, the modules describing the inputs and availability of N for leaching are presented, together with some results of predicted leaching at a national scale. The hydrological modules and spatial integration to predict

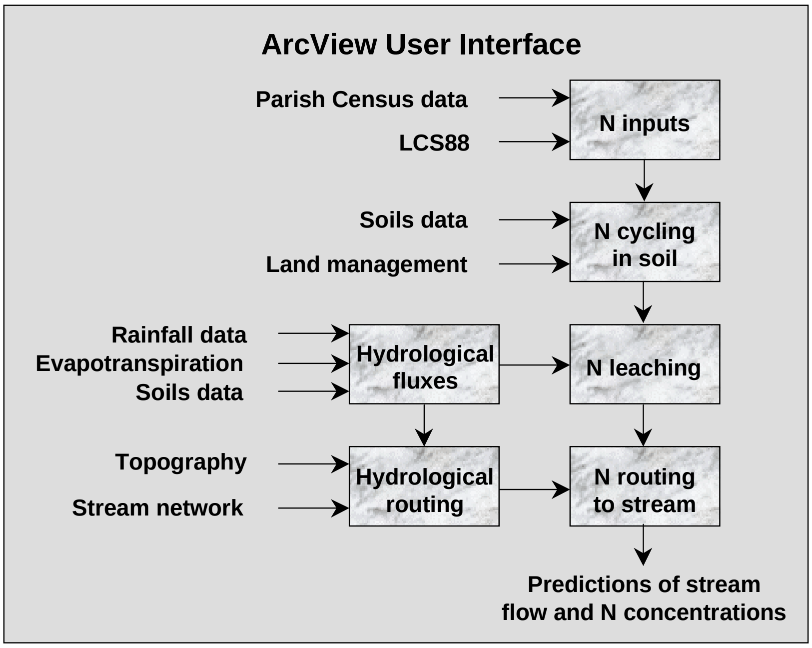

Fig. 1. *Structure of the NIRAMS model*

streamwater flows at a catchment scale are described in the subsequent paper (Dunn *et al*., 2004).

Land use and land cover databases and manipulation

The relationship between land use and nitrate pollution has been demonstrated many times, e.g. Burt and Haycock (1992), Puckett, (1995), Ferrier *et al*. (1998). As a precursor to the development of NIRAMS, a simple regression analysis was applied to 172 of the Scottish Environment Protection Agency (SEPA) stream water quality monitoring sites, to determine the relationship between stream nitrate concentrations and the proportion of broad land cover types within each catchment. The analysis showed a strong positive correlation between nitrate levels in the surface waters and the proportion of both arable land and cultivated land ($r = 0.867$ and 0.840 respectively, $p<0.001$) within the catchment. There was also a significant negative correlation with the proportion of semi-natural land and woodland ($r = -0.817$, $p<0.001$). These results justified developing a modelling methodology for N prediction based on an association between N leaching and land use.

The land use and the proportion of crop types within each catchment affect the amount of nitrate available for leaching and its spatial distribution. They also determine the locations and quantity of organic manures that are produced and subsequently returned to the land as a fertiliser and soil conditioner. The Land Cover of Scotland (LCS88) digital dataset (MLURI, 1993) provided historic data on the spatial distribution of five main land use types throughout Scotland. However, regional and annual differences in crop rotations and in the proportion of crops grown throughout Scotland also affect the amount of nitrate available for leaching. The Agricultural and Horticultural Census data collected annually by the Scottish Executive for each 'parish' administrative unit within Scotland are essentially coarse-scale spatial statistics of the areas of crops grown within each of these areas and also the areas of managed grasslands. By combining the Land Cover and the census data, the amount of nitrate available for leaching within each parish could be determined and distributed spatially within the catchment based on the five broad land cover categories.

LAND COVER OF SCOTLAND

A digital dataset of land cover types in Scotland was derived by interpreting 1: 24 000 scale aerial photographs taken in 1988 (and 1989); this Land Cover of Scotland (LCS88) digital dataset (MLURI, 1993) is one of the major environmental datasets in Scotland. A total of 126 individual land cover classes was identified. The minimum area mapped varied with land cover type; for the main land cover classes, this was 10 ha (or 2 ha in the case of woodlands).

For the purposes of this project, the 126 land cover classes were grouped into five broad classes based on their potential to generate nitrate from diffuse sources and on their potential loading of organic manures (both from excretion during grazing and from manure spreading). The classes identified were:

(1) arable agricultural land including rotational grassland;
(2) improved long ley grassland;
(3) rough grazing;
(4) semi-natural vegetation including semi-natural and commercial woodlands;
(5) urban areas (including disturbed ground such as quarries, bings and tips).

Although LCS88 is now a rather old dataset, it is being used in conjunction with the census data which show only a 5% variation in the total area of arable land between 1989 and 1999. It therefore seems unlikely that there has been significant change in the location of the broad land cover classes that are spatially assigned using LCS88.

AGRICULTURAL AND HORTICULTURAL CENSUS DATA

The Agricultural and Horticultural Census comprises statistics on the areal extent of individual crop types and livestock numbers. The data are collected from farmers and are aggregated to parish level. Twenty-seven different arable crops are included in the Census data together with five categories of grassland: rough grazing, grass for mowing or grazing depending on whether it is less than or more than 5 years old. The grassland is further sub-divided depending on whether the livestock unit is for dairy, beef or sheep rearing.

For the different arable crops and grasslands, balances of the N inputs and outputs from the system can be used to calculate the residual N available for leaching each year. The Hydrology of Soil Types (HOST) classification (Boorman *et al*., 1995) is also used to assign each location to a soil drainage class, to account for the effect of soil wetness on denitrification rates. This is described in greater detail later. The total contribution to residual N from arable agriculture for each parish is determined for each combination of arable land cover class and HOST class. These values are then distributed spatially across the catchment using both HOST and land cover spatial datasets. Values for the residual N in each 1 km^2 grid cell are calculated by combining inputs according to the % of each category within a cell.

Calculation of residual nitrogen

The simplified approach adopted in the development of NIRAMS is that the residual N at the end of the growing season determines the amount of N available for leaching over the subsequent autumn and winter. After the end of the summer, uptake of N by plants is minimal and hydrological conditions result in greater drainage of water from the soil root zone. Balances of inputs and outputs of N for each land use can be used to estimate the amount of residual N for each year. Water fluxes calculated within the hydrological component of NIRAMS (Dunn *et al.* ,2004) then determine the timing and amount of actual N leaching that occurs. The N balances are calculated in different ways for arable crops and grassland.

N BALANCES FOR ARABLE CROPS

For each arable crop the residual N is estimated by:

N residual = N fertiliser + N organic waste – N crop offtake + N net deposition + N mineralisation – N denitrification – N live weight gain

Values for the various components of the balance were derived for different crops from a range of literature sources and experimental data and are presented in Table 1. The inputs of N from organic waste to arable land depend on livestock numbers and grassland areas and are estimated at the Parish scale. The methodology for calculating these budgets is described below.

Fertiliser N

Fertiliser inputs were based on average values for Scotland derived from the British Survey of Fertiliser Practice (Fertiliser Manufacturer's Association, 1999)

N crop offtake

N offtake in crops was determined from a range of sources including Von Alt and Wiemann (1990), HGCA (1998), Vinten (1999), Chadwick (2000), and SEERAD (2000), The values used are shown in Table 1.

Net atmospheric N deposition

Exchange with the atmosphere can provide a significant net input (or loss) of nitrogen compounds from the plant-soil system. The main loss of reactive nitrogen to the atmosphere results from ammonia (NH_3) emissions, mainly from decomposition of livestock excreta and from fertilised crops. Other emissions as N_2 and nitrous oxide (N_2O) to the atmosphere occur via denitrification (described below). Inputs of nitrogen from the atmosphere result from dry

Table 1. Database of values for N balance (kg ha^{-1}) for arable crops

Crop Type	*N fertiliser*	*N crop offtake*	*N atmospheric deposition*	*Net organic N mineralisation*	*N live weightgain*
Winter wheat	202	173	10	70	0
Spring barley, Rye, Other stock feeding crops, Other crops	99	107	10	45	0
Winter barley	166	139	10	45	0
Oats	121	105	10	45	0
Seed potatoes	111	100	10	45	0
Early potatoes	135	85	10	45	0
Main potatoes	125	137	10	70	0
Spring oil seed rape, Beans, Peas	137	100	10	45	0
Winter oil seed rape	210	140	10	45	0
Linseed	0	Not available	10	45	0
Fodder roots, Turnips and Swedes	98	0	10	0	20
Fodder leaf, Rape	147	0	10	0	20
Brassica veg, Kale and Cabbage	140	72	10	45	0
Other veg, Vegetables for human consumption	Residual N = 68	-	-	-	-
Bare fallow land, Orchard fruit, Soft fruits	Residual N = 45	-	-	-	-

deposition of gases and particles and wet deposition in precipitation. These include inputs of both oxidised nitrogen (NOy) and reduced nitrogen (NHx). Although the net exchange with the atmosphere can vary depending on landuse, agricultural activity and other nitrogen sources, a first approximation was used for the NIRAMS balance calculations, based on average values for the UK. The average NO_x deposition for the UK is approximately 8 kg N ha^{-1} (NEGTAP, 2001) and it was assumed that the combined NO_x deposition and non-legume N fixation (algae and free living soil N fixers) was 10 kg N ha^{-1}. For ammonia deposition, it was assumed that the ammonia that is volatilised from livestock is deposited relatively locally, so that there is no net loss of N due to ammonia volatilisation from the farm waste N inputs to land. Based on spatial calculations for ammonia emissions and deposition carried out by Dragosits *et al*. (1998), this assumption may lead to slightly greater estimates of residual N in the south and east of Scotland and slightly smaller estimates in the west and north (Fig. 2).

Net N mineralisation

Net N mineralisation was estimated at 45 kg N ha^{-1} for all crops except winter wheat and potatoes. This was based on observations from zero N treatments on drained plots (Vinten, 1999). It was assumed that highly responsive arable crops such as winter wheat and potatoes would be grown on soils with greater mineralisation rates, typical of the first

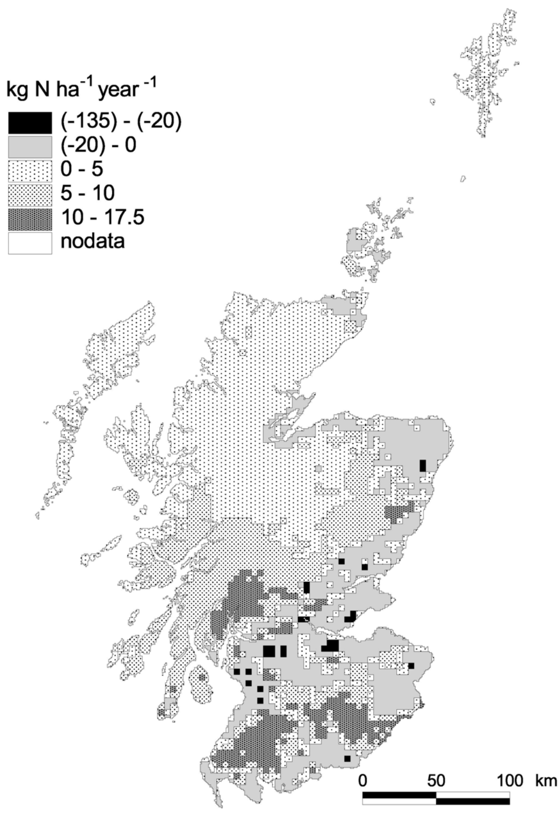

Fig. 2. *Balance between ammonia depositions and emissions for Scotland, 1995–97*

year after the ploughing of grass. From experiments on N release to cereals after long term grass, a figure of 70 kg N ha^{-1} was adopted (Vinten *et al*., 2002) This simplified representation does not account for differences in N processes as a function of soil type, temperature or the availability of soil N stores and as such will limit the potential of the NIRAMS model for predicting responses to changes in land management.

Denitrification

Denitrification rates are influenced strongly by soil wetness. Simulations were carried out by Vinten (1999) using the ANIMO model (Rijtema and Kroes, 1991) to estimate the proportion of fertiliser N lost by denitrification at sites with sandy loam and clay loam soils. This amounted to 21% in the sandy loam textured soils and 32% in the clay loam textured soils. A methodology was, therefore, developed for NIRAMS to group Scottish soils according to the proportion of fertiliser N that will be lost by denitrifcation. The HOST classification (Boorman *et al.*, 1995) was used to determine three main groups:

(1) those with freely draining profiles;
(2) those where the soils exhibit some degree of anaerobiosis in the subsoil (indicating the potential for periodically waterlogged conditions in the topsoil);
(3) those where the soils will be wet within the topsoil for substantial periods in the year.

The freely drained soils are assumed to show little denitrification while the periodically waterlogged soils, such as the sandy loam textured soils described above, are assumed to lose 20% of fertiliser N to denitrification. The remaining group, typified by the clay loam textured soil, is assumed to lose 35% of fertiliser N.

N BALANCES FOR GRASSLAND

Estimation of residual N for grassland cannot be treated as simplistically as for arable crops, because of the complex relationships between inputs and offtakes by livestock and the impact of winter housing on the N balance. An alternative method has been based on calculations of actual leached N from an application of the SOILN model (Wu *et al*., 1998) to an area of Dumfries and Galloway in south-west Scotland. The Dumfries and Galloway area has a high annual rainfall (typically > 2000 mm) and it is, therefore, reasonable to assume that, in these conditions, all residual N would be leached. This means that the calculated actual leached N can be used as a surrogate for residual N in other parts of the country.

The SOILN simulations involved calculation of average annual losses and N balances using 25 years of weather data with a daily modelling time step. For dairy herds, the quantities of N excreted in urine and faeces were estimated using values presented by Jarvis (1993) for both summer grazing and the winter-housed period. The stocking density was set at 2.7 Livestock Units (LU) ha^{-1} with half the grass area allocated to grazing, and the remaining area being cut for silage and receiving all the slurry produced during the winter-housed period. A typical fertiliser input of 270 kg N ha^{-1} for dairy grassland (Jarvis, 1993) was assumed. On the rest of the non-dairy grassland area, a stocking density of 2 LU ha^{-1}, typical for suckler beef herds, was assumed (Chadwick, 2000). This received fertiliser inputs of 140 and 116 kg N ha^{-1} on <5 and >5 year grass respectively, based

Table 2. Simulated actual nitrate N leaching from grassland for 3 soil drainage classes under different grassland categories, for weather conditions at Crichton Royal Farm, Dumfries.

Grassland type	*Livestock intensity*	*Livestock (units / ha)*	*Fertilizer input (kg / ha)*	*Excretal N input (kg /ha)*	*Leached N (kg / ha)* Freely drained	*Periodi-cally water logged*	*Prolonged water logging*
Grass for mowing (under 5 years old)	Dairy	2.7	270	270	57	38	19
Grass for mowing (over 5 years old)	Dairy	2.7	270	270	78	53	28
Grass for grazing (under 5 years old)	Dairy	2.7	270	270	78	57	35
Grass for grazing (over 5 years old)	Dairy	2.7	270	270	133	106	79
Grass for mowing (under 5 years old)	Beef+sheep	2	140	140	32	22	12
Grass for mowing (over 5 years old)	Beef+sheep	2	116	140	33	24	14
Grass for grazing (under 5 years old)	Beef+sheep	2	140	140	32	22	12
Grass for grazing (over 5 years old)	Beef+sheep	2	116	140	34	25	15

on national average data (SEERAD 2000). All the simulations were repeated for both long-term, permanent pasture and for newly re-seeded pasture (as would be found in a ley-arable rotation). The long-term pasture was assigned a large initial organic matter content of around 10%, to generate an approximately steady-state organic matter pool during the simulations. A smaller organic matter content of 3% was assigned to re-seeded pasture, to generate a gradual build up of the organic matter pool in the annual N balance. The results of the SOILN leaching simulations are summarised in Table 2 as a function of soil type.

Potential leaching from rough grazing is set at 2 kg N ha^{-1}, based on figures from DOE (1994).

BUDGETING FOR EXCESS ANIMAL EXCRETA

Inputs from animal excreta make a major contribution to the N budget of Scottish catchments. In addition to grazing animals, intensive livestock units for pigs and poultry also produce a significant amount of waste. Within NIRAMS the assumption has been made that excess animal waste is applied to the land in the parish within which it was produced.

Based on the detailed calculations of McTaggart *et al.* (2002), the following figures were used to estimate overall N excretion rates from livestock in kg N/animal:

dairy: 100 kg N/animal/yr on dairy farms
beef: 50 kg N/animal/yr on other farms
pigs: 8 kg N/animal/yr (including sows, fatteners and others)
sheep: 3.5 kg N/animal/yr (lambs included)
poultry: 0.6 kg N/bird/yr (including broilers, layers and others).

The figures include correction for various ages of stock. The estimates of residual N for grassland, described above, include typical waste returns for dairy and non-dairy farms. In addition to this, excess waste N production, over and above the standard inputs to grassland, may be generated within a parish. This excess waste must be accounted for within the broader scale simulations. The following procedure has been derived for NIRAMS to include excess waste in the N budget:

(1) Calculate the total N produced in excreta over the parish.
(2) Allocate this N successively, until all waste N has been used, to:
 Rough grazing at 10 kg N ha^{-1} (this is approximately equivalent to 0.2 beef cows per ha);
 Grass on dairy farms at 270 kg N ha^{-1};
 Grass on non-intensive farms at 140 kg N ha^{-1}.
 These figures are used in the standard calculations for residual N for grassland.
(3) Allocate any remaining waste N to arable land, including it as the value for *N organic waste* in the N balance equation.

Implicit in this calculation is the assumption that excreta are applied to land in the same parish in which it is generated. Table 3 gives some examples of these balance sheet calculations, with and without adjustment for excess waste, for four parishes in Scotland, based on 1995 census data. On an individual parish basis it is likely that some of the budgets will not be accurate because, in practice, there will be imports and exports of excess waste between parish areas. Errors are likely to be largest where parishes are small (parish areas vary from < 1 km^2 to > 1000 km^2 with a median area of 43 km^2 and a mean area of 88 km^2). Calculations of the budgets for organic waste on a national basis showed a net annual loss in organic N of 2 kg N ha^{-1}, which is acceptably small. By including the animal waste in the balance equation the assumption is made that the timing of application is not significant, and that no major losses occur as a result of direct wash off. This assumption is recognised to have its limitations.

OTHER N INPUTS

The focus of NIRAMS is on prediction of concentrations and fluxes of nitrate draining from agricultural land. However, in order to evaluate the performance of the model at catchment scales, a simple estimate of N losses from urban areas has been included. This applies a uniform addition to the residual N balance throughout the year of 20kg ha^{-1} y^{-1} for all areas that are defined as having an urban land cover. In some rural areas losses of N from domestic septic tanks might also be expected to contribute detectable loads to streams. However, the availability of geo-referenced data to define locations of septic tanks is poor and therefore these inputs have not been accounted for in NIRAMS.

Calculation of N leaching

The residual N left in the soil at the end of the growing season is assumed to be available for leaching. The time sequence of the leaching loss is an important factor in determining the temporal variability and range of variation in stream N concentrations.

Various possible models for describing the leaching in NIRAMS were considered, including those suggested by Lord and Anthony (2000), and a combination of the vertical

Table 3. Summary of excess excreta calculations, showing effect on residual N for arable land for 4 parishes in Scotland, based on 1995 census data.

	Area(ha)	*Excretal N(kg / ha)*	*Excretal N(kg)*
CRAIGIE, AYRSHIRE:			
Rough Grazing	44	10	438
Improved grassland, dairy	758	270	204668
Improved grassland, non-dairy	1083	45	49208
Arable	380	0	0
Total	1885		254313
Arable residual N (no excess waste) = 43 kg / ha			
Arable residual N (waste adjusted) = 43 kg / ha			
DYCE, ABERDEENSHIRE:			
Rough Grazing	97	10	971
Improved grassland, dairy	39	270	10474
Improved grassland, non-dairy	612	106	64893
Arable	322	0	0
Total	1070		76338
Arable residual N (no excess waste) = 37 kg / ha			
Arable residual N (waste adjusted) = 37 kg / ha			
BARRY, ANGUS:			
Rough Grazing	7	10	74
Improved grassland, dairy	0	-	0
Improved grassland, non-dairy	663	140	92806
Arable	451	155	69841
Total	1121		162647
Arable residual N (no excess waste) = 47 kg / ha			
Arable residual N (waste adjusted) = 202 kg / ha			
WHITSOME, BERWICKSHIRE:			
Rough Grazing	0	-	0
Improved grassland, dairy	41	270	11026
Improved grassland, non-dairy	166	140	23207
Arable	1453	4.5	6682
Total	1660		40915
Arable residual N (no excess waste) = 77 kg / ha			
Arable residual N (waste adjusted) = 81 kg / ha			

leaching function described by Burns (1975) with a steady state model of horizontal flow to tile drains (Van Ommen, 1985). Tests on these approaches suggested that they might not be the most appropriate for this application. For example, the leaching curve presented by Lord and Anthony (2000) would predict that there would be zero nitrate leaching from the soil profile after 300–400 mm of drainage. Since net precipitation (rainfall minus evapotranspiration) is well in excess of this in the wetter arable areas of Scotland, and as observations from drained plots show nitrate concentrations around 3 mg l^{-1} at the end of the winter, such loss rates are inappropriate for Scottish conditions. The Burns / Van Ommen model provided a better prediction of N leaching when compared with experimental plot data. However, the structure of the model would be cumbersome to implement within a GIS framework.

NLEAP MODEL

As an alternative to these methods, the use of the simple exponential function for describing N leaching as used in NLEAP (Shaffer *et al*., 1994) was investigated.

The NLEAP function takes the form:

$$NL_t = NAL_t \times (1 - \exp(-KNL \times (WAL_t / SATC)))$$

where NL_t is the N leached in week t (kg ha^{-1}), NAL_t is the N available for leaching (kg ha^{-1}), KNL is the leaching coefficient (-), WAL_t is the water available for leaching (mm) and *SATC* is the soil saturated capacity (mm).

Within the NIRAMS water balance model, the hydrological fluxes are separated into three components identified as overland flow (*OF*), sub-surface flow (*SSF*) and groundwater flow (*GWF*) as described in Dunn *et al.*, (2004). *WAL* is the total hydrological flow from a cell, with the exception of overland flow. Hence:

$$WAL_t = GWF_t + SSF_t$$

NAL at the end of the growing season is equivalent to the residual N. *NAL* is subsequently depleted with time as N is leached:

$$NAL_t = NAL_{t-1} - NL_{t-1}$$

Shaffer *et al*. (1994) assigned a value of *KNL*=1.2 for all soils where the soil water content at approximately –15 000 cm was more than 0.1 cm^3 cm^{-3}. In practice, this applies to virtually all soils in Scotland apart from soils with loamy sand or sand texture where the average retained water capacity at –15000 cm was calculated as 0.08 cm^3 cm^{-3}. Preliminary testing of the behaviour of the NLEAP function, outwith the remainder of the NIRAMS code, suggested that a value of *KNL*=1.2 may result in over-prediction of the leaching rate. A value of *KNL*=0.7 was found to give a better fit to experimental data from a drained plot with sandy-loam soil. Further sensitivity analysis was therefore carried out during testing of the NIRAMS model, to establish the effect of *KNL* on N leaching. *KNL* is currently applied using a uniform value across a catchment. Further work at the sub-catchment scale may show how the value of *KNL* can be linked to individual soil types.

The N leached from the soil is partitioned into two components in direct proportion to the volumes of groundwater and sub-surface runoff. Thus:

$$NGW_t = NL_t \times GWF_t / (GWF_t + SSF_t)$$

$$NSS_t = NL_t \times SSF_t / (GWF_t + SSF_t)$$

where NGW_t (kg ha^{-1}) is the N added to the groundwater N store at time t and NSS_t (kg ha^{-1}) is the N added to the sub-surface store. The subsequent hydrological routing of these N stores to the stream system is described in Dunn *et al*. (2004).

Model Predictions

RESIDUAL N CALCULATIONS

The NIRAMS model has been applied using five years of historic land use data to calculate the residual N following each of the growing seasons from 1989 until 1993. The average residual N values calculated for the whole of Scotland vary little over the five-year period, ranging from 11.2 kg N ha^{-1} yr^{-1} to 11.7 kg N ha^{-1} yr^{-1}. A map of the calculated residual N for 1990 is shown in Fig. 3. This output demonstrates the strong regional variability in agricultural practice across Scotland. The greatest residual N values occur in the east where arable agriculture is most common and where residual N is typically between 30 and 60 kg N ha^{-1} yr^{-1}. Large areas of rough grazing in the north and north west account for the low N inputs over much of the country. Intermediate values of 15 to 30 kg N ha^{-1}, in the south and south west, correspond with areas of dairy farming.

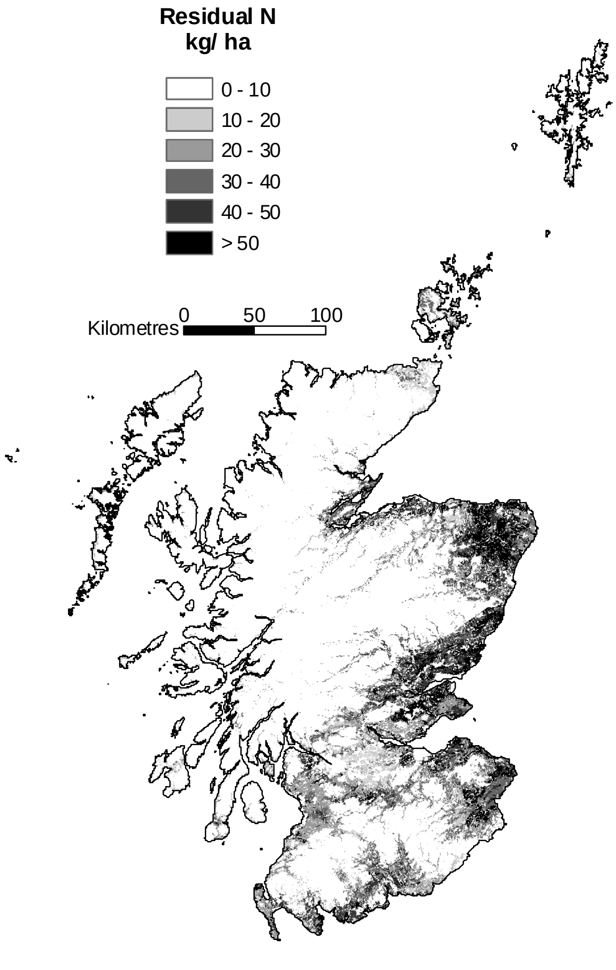

Fig. 3. *Map of calculated residual N for Scotland following the 1990 growing season*

LEACHED N

Predictions of actual leached N for the same five-year period have been made using weekly rainfall and evapotranspiration data to drive the NIRAMS water balance model. The derivation of the spatial meteorological inputs is described in Dunn *et al*. (2004). Figure 4 shows a map of the total leached N for the whole country summed over a full year from autumn 1990 to autumn 1991; the greatest leaching occurs in the areas that had greatest residual N levels. However, the variability in actual leached N is less than the variability in residual N, because a lesser proportion of residual N is leached in some of the areas with greatest residual N in the east of the country. Figure 5 highlights this, showing the amount of actual leached N as a percentage of the residual N. The difference in proportions of N leached reflects the strong trend in decreasing availability of water resources from west to east across Scotland. In some of the arable areas in the east, the excess of residual N over that which is leached is as great as 25 kg N ha^{-1} and this excess is likely to be immobilised in the soil. This may have contributed to the gradual increase in soil N stores that have been observed in the last 50 years.

As an illustration of the variability in N leaching around the country, average data have been calculated for eight diverse catchments, geographically distributed around Scotland (Fig. 6). The physical characteristics of these catchments are summarised in Table 4. The Ythan, Eden and Tyne catchments are predominantly arable areas, all draining eastwards to the North Sea. The North Esk and Earn are mixed catchments with a significant area of rough grazing in the upper areas of the catchments, as well as arable areas and permanent grassland in their lowlands. The Urr and Irvine catchments both have dairy farming as the predominant form of agriculture and drain to the south west and west respectively. The Carron catchment in the north west of the country is a typical upland catchment with few agricultural inputs. Table 5 summarises the predicted residual and leached N figures averaged across each of these

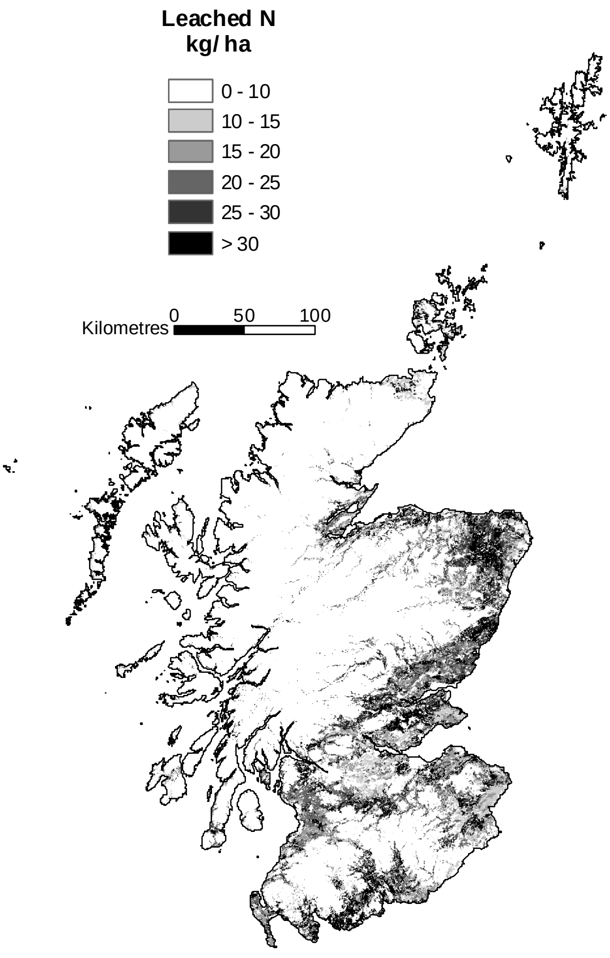

Fig. 4. *Map of calculated leached N for Scotland summed over a year starting from autumn 1990*

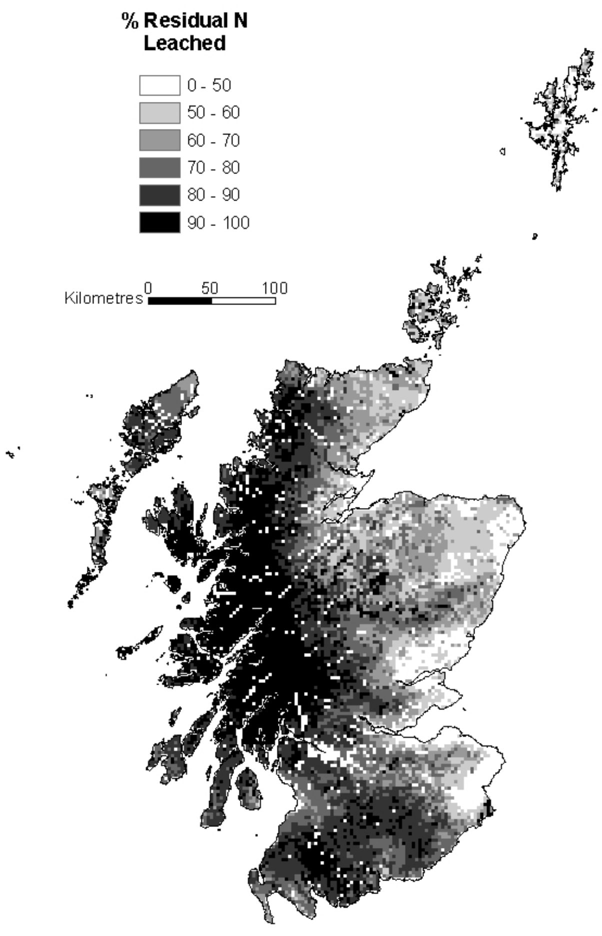

Fig. 5. *Calculated percentage of residual N leached during a year starting from autumn 1990*

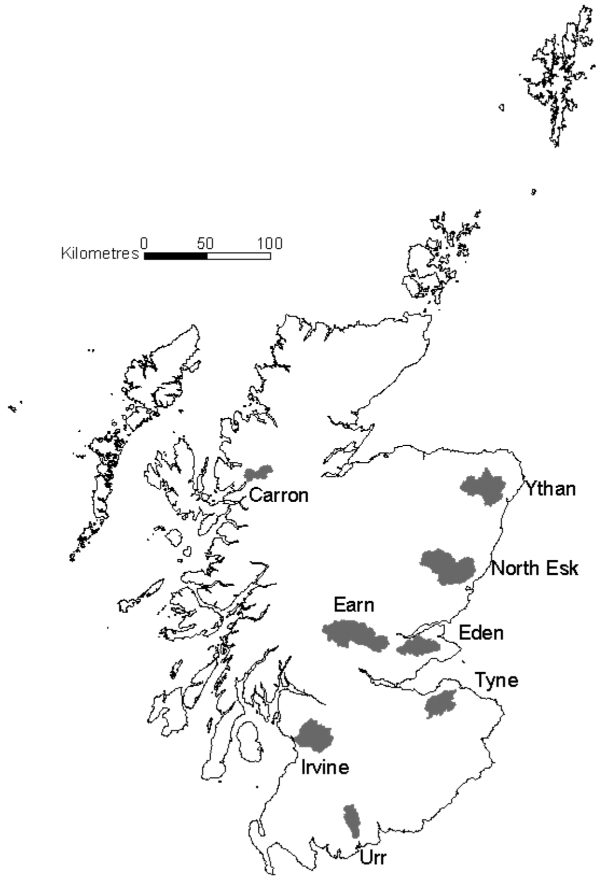

Fig. 6. *Locations of NIRAMS test catchments*

Table 5. Predicted residual N and leached N for autumn 1990–autumn 1991 for eight catchments

Catchment	*Residual N kg ha^{-1}*	*Leached N kg ha^{-1}*	*% Leached*
Ythan	56	29	52
N Esk	18	11	61
Earn	21	16	76
Eden	48	28	58
Tyne	33	20	61
Urr	16	14	88
Irvine	21	16	76
Carron	2	2	100

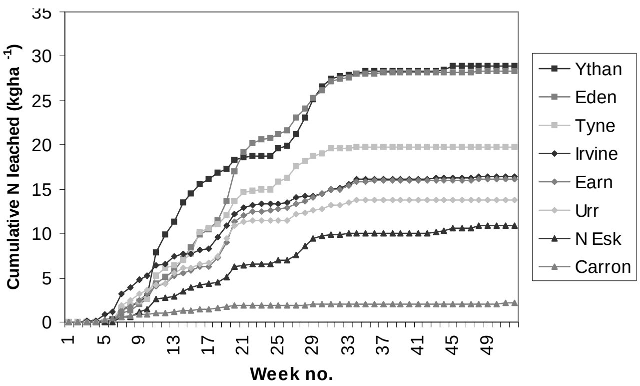

Fig. 7. *Predicted cumulative N leached for 8 catchments from autumn 1990 to autumn 1991*

catchments for 1990–1991. These figures highlight the much greater residual N levels in the predominantly arable catchments, as well as demonstrate the influence of climate on the actual leached N. The North Esk catchment has greater residual N than the Urr catchment, but a lesser actual leached N total.

Figure 7 shows the timing of N leaching for the eight catchments in terms of the predicted cumulative N leached, starting from the end of August 1990. In the first 10 weeks, more leaching occurs in the Irvine and Urr catchments than in the drier east-coast catchments. By the end of March, the leaching curves have largely flattened off, and there are only

Table 4. Summary of characteristics of NIRAMS test catchments

Catchment	*Area km^2*	*Av. ann. rain mm*	*% Arable*	*% Improved pasture*	*% Rough grazing*	*% Freely drained soil*	*% Poorly drained soil*
Ythan	526	826	88	5	5	69	18
N Esk	740	1074	26	8	57	37	56
Earn	738	1397	24	6	56	41	36
Eden	318	799	75	5	7	62	25
Tyne	313	713	65	13	12	31	54
Urr	200	1340	0	48	32	33	66
Irvine	480	1228	0.4	66	14	10	89
Carron	149	2620	0	1	86	39	61

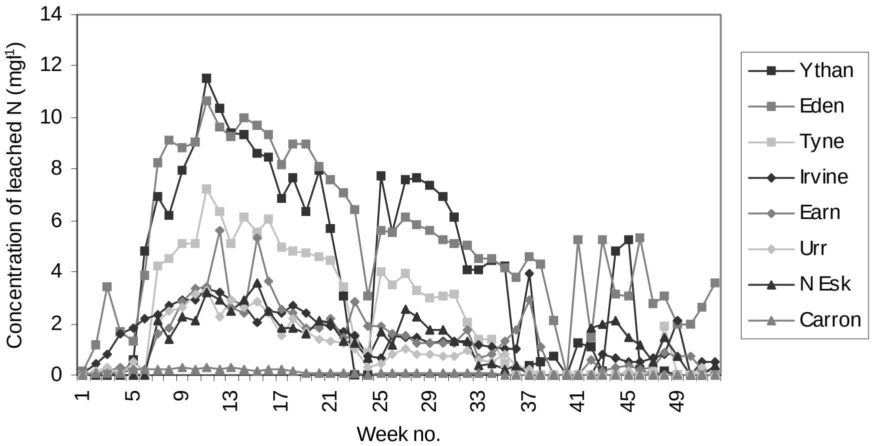

Fig. 8. *Concentrations of N leached for 8 catchments from autumn 1990 to autumn 1991*

small losses from then until the end of the summer. For most of the catchments, low hydrological flows rather than the supply of N limit leaching in spring and summer.

The concentrations of N in leachate are plotted as weekly time-series in Fig. 8. This shows that there is a gradual decline in concentration through the winter. However, when leaching does occur during the spring and summer, concentrations can still rise to between 2 and 5 mg l^{-1}. These results are compatible with observations that drainage water from experimental arable plots in central Scotland showed nitrate concentrations of around 3 mg l^{-1} at the end of winter (Vinten, 1999).

Discussion and conclusions

A pragmatic methodology for predicting the amount of N leaching from agriculture in Scotland has been developed and integrated within a national scale model. The model is based on national datasets and has minimal requirements for parameter calibration. As such this makes it appropriate for use as a simple modelling tool for identifying areas that are likely to be susceptible to N pollution. The predictions of leaching made by the model for a five-year simulation demonstrated the high variability in N leaching across Scotland. The results showed a clear geographical pattern to the N leaching with the greatest leaching occurring in the east of the country and corresponding to the areas of arable agriculture. High levels of N leaching were also predicted in parts of the south and west of Scotland where there are large numbers of livestock. However, concentrations of leachate would be lower in these areas because of the higher precipitation inputs that cause greater dilution of the N. The simulations also suggested that the losses of N in the areas with greatest residual N are not directly proportional to the amount of residual N, because they coincide with lower rainfall. However, these areas are most at risk from deteriorating water quality in the longer-term because of the gradual accumulations of residual N within the soil. Depending on other soil conditions, the accumulating N store may lead to increases in mineralisation resulting in a higher residual N under the same agricultural management. Spatial outputs of the type that have been produced here are potentially of great value in relation to the WFD as they provide a methodology for quantifying the relative degree of risk of nitrate pollution in different parts of Scotland. In this context, a simplified version of NIRAMS has recently been adopted and integrated into a screening tool for diffuse pollution being developed to assist SEPA with production of a River Basin District characterisation report as part of the implementation of the WFD (Betson *et al*., 2004).

The NIRAMS model is currently limited in its ability to predict long-term changes in leaching losses because there is no carry over of 'unleached' residual N from one year to the next. To incorporate such a facility it would be necessary to include a more explicit temporal representation of some of the N cycle processes, such as immobilisation and mineralisation, and this would clearly add to the model's complexity. However, for more comprehensive studies of the impacts of management changes on future N losses, it would be advantageous to include this capability.

The calculations of residual N have been based on the best data available from the literature and from experimental studies in Scotland. Inevitably a degree of uncertainty is associated with these values, particularly where data have been available only from a single source. Values for the residual N in grassland were particularly difficult to establish, due to the complexity of the N cycle on grazed land. Because of this, a rather complex algorithm has been necessary to carry out organic waste balances within each parish area. The algorithm that has been used is not perfect, because there is no facility to reduce the residual N below expected typical values for improved grassland, in situations where the livestock numbers in a parish generate below average N inputs to the system. In practice, the error is likely to be small because of a compensating reduction in the crop offtake where the N inputs are lower. Due to the available resolution of spatial data (in particular the Agricultural and Horticultural Census data) the model is unlikely to operate successfully under application to catchments of less than around 30 km^2 in size. Other limitations in terms of the N budget include neglecting losses from septic tanks and no account of leaching of dissolved organic nitrogen (DON). It is believed that this will not cause a significant error in terms of the overall budgets at a catchment scale.

As the model stands it can be used to study simple effects of land use change. The model simulations carried out were based on historical records of land use from the Agricultural and Horticultural Census data. However, within the NIRAMS model interface a facility has been incorporated that permits the user to input their own figures to define areas of land use in different parishes. Using this facility, scenarios can be developed to study the effect on leaching losses of simple changes in cropping patterns. The effect of reducing livestock numbers on excess excreta can also be included, easily. However, more complex management changes involving reductions in fertiliser applications require recalculation of the N balances for different crops and are, therefore, more complex to implement using the basic model.

The leaching calculations demonstrated the importance of water availability in determining N losses, both through the temporal control of the loss throughout the winter, and in terms of the total amount of the residual N that is actually leached. Hydrology is of further importance in controlling the transport of N via various flowpaths to surface waters. Dunn *et al*., (2004) describe how the hydrological influences have been incorporated in NIRAMS and demonstrates the application of the full modelling system at the catchment scale.

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References

Betson, M., Anthony, S., Lord, E., Dunn, S., Lilly, A., Towers, W., De Groote, J., Struve, J., Tagg, A., Abbott C., Lewis, D., MacDonald, J., Badger, R., Ferris, N. and Ward, B., 2004. A GIS decision support screening tool for diffuse pollution in Scotland and Northern Ireland. In: *Agriculture and the Environment: Water Framework Directive and Agriculture*", D. Lewis and L Gairns (Eds.) Proceedings of SAC and SEPA Biennial Conference Edinburgh, 24-26 March 2004. 107–113

Boorman, D.B., Hollis, J.M. and Lilly, A., 1995. Hydrology of soil types: a hydrologically-based classification of the soils of the United Kingdom. *Institute of Hydrology Report No. 126*. Centre for Ecology and Hydrology, Wallingford, Oxon.UK. 137pp.

Burns, I.G., 1975. An equation to predict the leaching of surface applied nitrate. *J. Agr. Sci.,* **85**, 443–454.

Burns, I.G., 1980. A simple model for predicting the effects of winter leaching of residual nitrate on the nitrogen fertilizer need of spring crops. *J. Soil Sci.*, **31**, 187–202.

Burt, T.P. and Haycock, N.E., 1992. Catchment planning and the nitrate issue - a UK perspective. *Prog. Phys. Geog.,* **16**, 379–404.

Chadwick, L., 2000. *The farm management handbook*. Scottish Agricultural College, Edinburgh, UK. 518pp

Department of the Environment, 1994. Impacts of Nitrogen Deposition in Terrestrial Ecosystems. *Report of the UK Review Group on Impacts of Atmospheric Nitrogen*, C.E.R. Pitcairn (Ed.). Centre for Ecology and Hydrology, Bush Estate, Edinburgh, UK. 314pp.

Dragosits, U., Sutton, M.A., Place, C.J. and Bayley, A. 1998. Modelling the spatial distribution of ammonia emissions in the UK. *Environ. Pollut.* (Nitrogen Conference Special Issue). **102**, S1, 195–203.

Dunn, S.M., Lilly A., DeGroote, J. and Vinten, A.J.A., 2004. Nitrogen Risk Assessment Model for Scotland: II. Hydrological transport and model testing, *Hydrol. Earth Syst. Sci.,* **8,** 205–218.

European Community, 1991. Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources. *Official Journal* **L375**.

European Community, 2000. Directive 2000/60/EC of the European Parliament and of the Council, establishing a framework for Community action in the field of water policy. *Official Journal* **L 327**.

Ferrier, R.C., Littlewood, I.G., Hurst, D. and Watts, C.D., 1998. *Review of the Harmonised Monitoring Scheme (Scotland) 1974-1994*. Macaulay Institute, Aberdeen, UK. 129pp.

Fertilizer Manufacturer's Association, 1999. *The British survey of fertilizer practice: fertilizer use on farm crops, UK.* The Fertilizer Manufacturers Association, Peterborough, UK.

HGCA, 1998. Cereals statistics, 1998. *The farmer's handbook*. Kluwer-Harrap Address?.

Jarvis, S.C., 1993. Nitrogen cycling and losses from dairy farms. *Soil Use Manage.* **9**, 99–105.

Johnes, P., 1996. Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach, *J. Hydrol*., **183**, 323–349.

Lord, E.I. and Anthony, S.G., 2000. MAGPIE: A modelling framework for evalauating nitrate losses at national and catchment scales. *Soil Use Manage.,* **16**, 167–174.

Marsden, M.W. and MacKay, D.W., 2001. Water quality in Scotland: the view of the regulator. *Sci. Total Environ.*, **265**, 369–386.

McTaggart, I.P., Sinclair, A.H. and Smith, J., 2002, *Nitrate Vulnerable Zone Action Programme Regulatory Impact Assessment Report*, Scottish Executive Environment and Rural Affairs Department, Edinburgh, UK.

MLURI., 1993. *The Land Cover of Scotland 1988*. Macaulay Land Use Research Institute. Aberdeen, UK..

National Expert Group on Transboundary Air Pollution (NEGTAP), 2001. Transboundary Air Pollution: *Acidification, Eutrophication and Ground Level Ozone in the UK*. D. Fowler (Ed.).

Neitsch, S.L., Arnold, J.G., Kining, J.R., Williams, J.R. and King, K.W., 2002. *Soil and Water Assessment Tool Theoretical Documentation v2000*, Grassland Soil and Water Research Laboratory, Agricultural Research Services, Texas, USA. 506pp.

Puckett, L.J., 1995. Identifying the major sources of nutrient water pollution, *Environ. Sci. Technol*., **29**, 408–414.

Rijtema, P.E. and Kroes, J.G., 1991. Some results of nitrogen simulations with the model Animo. *Fertilizer Research.* **27**, 189–198.

SEERAD, 2000. Economic Report on Scottish Agriculture. 2000 Edition. *Government Statistical Service*. 82pp.

SEERAD, 2003. Maps of areas designated as Nitrate Vulnerable Zones in Scotland under the Nitrates Directive (91/676/EC), Scottish Executive, Edinburgh, UK.

SEPA, 1999. Improving Scotland's Waters, *Scottish Environment Protection Agency*, Stirling, UK.

SEPA, 2000. Annual Report, *Scottish Environment Protection Agency*, Stirling, UK.

Shaffer, M.J., Wylie, B.K., Follet, R.F. and Bartlett, P.N.S., 1994. Using climate/weather data with the NLEAP model to manage soil nitrogen. *Agr. Forest Meteorol.* **69**, 111–123.

Smith, R.A., Schwarz, G.E. and Alexander, R.B., 1997. Regional interpretation of water-quality monitoring data. *Water. Resour. Res*., **33**, 2781–2798

Van Ommen, H.C., 1985. Systems approach to an unsaturated-saturated groundwater quality model, including adsorption, decomposition and bypass. *Agr. Water Manage.* **10**, 193–203.

Von Alt, D. and Wiemann, F., 1990. Stickstoff im Erntegut und in den Ernteruckstanden von Gemuse. *Gemuse* **7,** 352–356.

Vinten, A.J.A., 1999. Predicting nitrate leaching from drained arable soils derived from glacial till. *J. Environ. Qual.,* **28**, 988–996.

Vinten, A.J.A., Ball, B.C., O'Sullivan, M.F., Howard, R., Wright, F., Ritchie, R. and Henshall, K., 2002. The effects of cultivation method and timing, previous sward and fertilizer level on subsequent crop yields and nitrate leaching following cultivation of long term grazed grass and grass-clover swards. *J. Agr. Sci.* (Cambridge) **139**, 231–243.

Whitehead, P.G., Wilson, E.J. and Butterfield, D., 1998. A semi-distributed Integrated Nitrogen model for multiple source assessment in Catchments (INCA): Part I – model structure and process equations. *Sci. Total Environ*., **210**, 547–558.

Wu, L., McGechan, M.B., Lewis, D.R., Hooda, P.S. and Vinten, A.J.A., 1998. Parameter selection and testing the soil dynamics model SOILN. *Soil Use Manage.,* **14**, 170–181.

Young, R.A., Onstad, C.A., Bosch, D.D. and Anderson, W.P., 1989. AGNPS: A non-point-source pollution model for evaluating agricultural watersheds. *J. Soil Water Conserv.,* **44**, 168–173.